Cost effectiveness of endangered species management: the kokako (*Callaeas cinerea*) in New Zealand

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Abstract: Expenditure on endangered species management is increasing greatly, on a global basis. Managers need tools to evaluate the performance of endangered species programmes because there will always be more demand for resources than there are available. Cost Effectiveness Analysis (CEA) is used here to evaluate the performance of the kokako (*Callaeas cinerea*) recovery programme. This species is being managed at a number of sites in New Zealand and analysis shows a large variation in costs and effectiveness between these sites. Cost Effectiveness Analysis provides a tool to allow managers to better predict where resources should be invested to most cost-effectively achieve their conservation targets, in this case recovery of an endangered species. Issues of lack of reliable cost data and ongoing policy problems limit the potential of economics to contribute to improved conservation management of threatened species in New Zealand.

Keywords: Cost Effectiveness Analysis; endangered species; kokako; management; New Zealand.

Introduction

Over 2000 plant and animal species are considered threatened in New Zealand (Hitchmough, 2002). New Zealand's Environment 2010 Strategy (Ministry for the Environment, 1994) identified that "… protecting indigenous habitats and biological diversity …" was the highest priority environmental objective. However, by attempting to protect or restore all indigenous species and natural habitats when we do not have the resources to do so, we risk sacrificing species that can be saved. Equally, setting goals for individual species requires choices to be made about management actions. Ecology and economics should lie at the heart of threatened species management. Ecology is vital, since we need to know what is most important biologically, and what is ecologically possible to achieve. Economics is vital because, with a finite budget, we need to know the cost utility of actions (see Cullen *et al*., 1999, 2001, 2002) on an inter-species basis and the cost effectiveness of alternative actions on an intra-species basis. The New Zealand Department of Conservation's (DOC) strategic business plan has indicated that DOC must become more proficient in evaluating performance and making transparent resource allocation decisions (Department of Conservation, 1998).

A basic objective to strive for in setting conservation priorities is to 'maximise biodiversity conserved from a given budget' (Moran and Pearce, 1998). This principle of maximum conservation is intuitively appealing, but it is unworkable unless there is a clear goal that conservation agencies are trying to achieve (Metrick and Weitzman, 1998). To estimate the optimal mix of conservation investments and evaluate the performance of conservation managers it is fundamental to specify the criterion on which they are to be assessed (Metrick and Weitzman, 1998). Constructing an objective function that can be used to answer questions about the relative cost effectiveness of conservation efforts has proved difficult.

Metrick and Weitzman argue that in order to set priorities for maintaining and increasing diversity, decision-makers must (1) specify what it is they are aiming to maximise, and (2) clearly state the optimisation problem in a cost effectiveness ranking formula. Nature conservation typically requires budgetary trade-offs. Decision-makers have to decide what programmes and projects they are going to pursue each financial year from a shopping list of programmes and projects. The objective is to estimate the payoff from conservation so that it is possible to maximise conservation returns from a range of potential programmes for a given level of risk, although this is complicated by the dynamics of the conservation management problems faced by programme managers.

Management priorities for threatened species conservation

Conservation managers in New Zealand can protect or enhance natural assets through various techniques or methods such as aerial pest control, commercial hunting, fencing, habitat manipulation and strategies such as mainland island management and offshore island protection (Parkes, 1996). In making choices for species and habitat management hard decisions must be made on (1) the means of protection or enhancement, and (2) the relative desirability of conservation goals or ends.

If conservation policy analysis is to be rational it is necessary to estimate the cost effectiveness of policy goals or programmes. For example, has it been more cost effective to invest in kokako (*Callaeas cinerea*) rather than in the black stilt (*Himantopus novaezealandiae*), taiko (*Pterodroma magentae*) or grand skink (*Leiolopisma grande*)? How cost effective has kokako conservation been at different sites in the North Island? Given that programmes are in competition for the same limited budget they must be compared according to their cost effectiveness.

Aims

The purpose of this research is to assist with the development of an economic evaluation framework for estimating the productive efficiency of threatened species management. We report how we have used Cost Effectiveness Analysis to evaluate conservation management projects for a single species, the kokako. We use one output or effectiveness measure for estimating the productive efficiency of management. This measure allows us to easily compare alternative courses of action designed to achieve that output. Our approach is especially useful where analysts must be concerned with how conservation efforts are performing in terms of a taxon overall (e.g. kokako) and the status of meta-populations at individual managed sites (e.g. kokako at Kaharoa, Mapara and Hunua).

Crucial to any economic analysis of nature conservation management actions is the choice of the unit of measurement. Species, populations, genes, ecosystems, communities and landscapes could all be used. However, species offer the most practical unit for conservation evaluation because they are much more tangible and easier to measure than are other units, e.g. communities or ecosystems, which are more complex and difficult to define (Goldstein, 1999; Noss, 1990). A substantial amount of DOC's efforts have also concentrated on single species. In 1998/99, out of a total appropriation of $161.8 million (GST exclusive), DOC spent $24.6 million on species conservation and a further $34.2 million on plant and animal pest control, which implicitly or explicitly is undertaken to protect species and habitats (Department of Conservation, 1999a). By 1999 a total of 31 species recovery plans had been published by DOC, which compares with only 13 ecosystem restoration projects, including 6 mainland islands projects (Saunders, 1999; Saunders and Norton, 2001).

The kokako is one of two extant species in the endemic New Zealand wattle bird family. It is a territorial forest bird species, formerly found in both the North and South Islands of New Zealand. However, the South Island race (*Callaeas cinerea cinerea*) may now be extinct, and the range of the North Island race (*Callaeas cinerea wilsoni*) is greatly reduced as a result of habitat loss and of predation (Innes and Flux, 1999). Remaining native forests are mostly protected under the statutory control of the Department of Conservation. However, the threats posed by introduced predators, mostly possums (*Trichosurus vulpecula*) and ship rats (*Rattus rattus*), are an ongoing problem and are the main focus of the North Island kokako recovery plan (Innes and Flux, 1999). The recovery plan has a 20-year goal "to improve the status of Kokako from endangered, by restoring the national population to ca. 1,000 pairs by the year 2020, in sustainable communities throughout the North Island." As well, "in order to attain the stated goal of this plan, we state 23 key sites which represent the necessary minimum management sites required to improve the status of kokako by 2020" (Innes and Flux, 1999). With management planned for 23 sites and predator control being the primary management requirement, it is appropriate to compare the cost effectiveness of management between sites to help decide where the best conservation return on investment has been occurring. Such knowledge might help managers decide where future investment should occur.

Cost Effectiveness Analysis

Cost Effectiveness Analysis (CEA) can be used to find the least cost means to meet a conservation objective using a single measure of effect. It differs from Cost Benefit Analysis because the managed outputs are usually measured in non-monetary units such as pest kill, abundance of species, number of habitat units protected or species survival probability. The analysis requires data on each management alternative's output and cost. Specifying the output is crucial because this will determine what alternatives can be included in the analysis. For example, if the effectiveness measure is pest kill, then only pest control operations can be included in the analysis. If many alternatives are being evaluated CEA may involve an optimisation procedure for identifying the actions which are the most cost effective.

Cost Effectiveness Analysis can also be used to assess the costs of different goals for a threatened species recovery plan. Several North American researchers have used stochastic population models to estimate the marginal costs of achieving various survival probabilities under alternative recovery plans (Haight, 1995; Hyde, 1989; Montgomery *et al.,* 1994). Although these studies have focused on the economics of the distributional impacts to society of conservation management (employment, earnings, profits), the methodologies employed are also applicable to the trade-offs between population size and direct cost.

Methods

Cost Effectiveness Analysis commonly measures the cost of programmes or projects in dollars but the outputs in non-monetary units. The unit of output or effectiveness measure should be derived from the goal one is trying to achieve. Since the principal goal stated in the North Island kokako recovery plan is 'additional number of male/female pairs', this is the best output measure for a CEA, at any of the managed sites. We recognise that the recovery plan aims to manage kokako at 23 sites, as well as achieving the goal of 1000 pairs by 2020. We discuss in a later section the role of the '23 sites' goal. The gain in the number of kokako pairs is a final rather than intermediate output. Knowledge obtained, such as 'managers have confidence that restoration is possible on the mainland', is an intermediate output and not the criterion that should be used. Of course, an improved knowledge base, via research-by-management for example, may lead to more birds in the future. By measuring the payoff in terms of final outputs one can get a much more accurate picture of what is being achieved with a given amount of resources.

To estimate the cost effectiveness of kokako conservation at sites throughout the North Island it is necessary to specify a simple equation for ranking kokako projects according to their cost effectiveness. In developing this equation four inter-related economic terms used throughout the remainder of the paper require definition. Discounting is a tool used to express the benefits that occur in different periods of time, either future or past, in a common metric. That metric is most commonly dollars, as for costs in this equation, but can also be other countable benefits such as the female/male pairs recorded here as measures of effectiveness. When discounted, the common metric used is referred to as Present Value, and future or past values are converted to a value in today's terms. This conversion is undertaken using the Discount Rate, commonly the rate at which governments pay to borrow money. We have chosen a discount rate of 6%. An amortisation amount is an annual payment that will meet the total costs of a given project, say an endangered species management project, over the project life.

A cost effectiveness formula for kokako can be expressed as follows:

$$PAYOFFk = \sum_{t=0}^{T}\left[\frac{K_n - K_i}{(1+d)^t}\right] / \sum_{t=0}^{T}\left[\frac{C_t}{(1+d)^t}\right]$$

where:

PAYOFFk is the change in the number of discounted male/female kokako pairs per discounted conservation dollar spent on kokako protection at a site.

K_n is the number of female/male pairs at the end of the planning period.
K_i is the number of female/male pairs at the beginning of the operational period.
C_t are the direct costs of protection at a site in year t.
d is the discount rate.

To measure the effectiveness of kokako management since 1989, we obtained from kokako recovery group members' annual data on number of male/female pairs each year from 1989 to 1998 at six managed sites—Rotoehu, Mapara, Hunua, Otamatuna, Mataraua and Kaharoa. Kokako are believed to reside in about 15 sites in the upper North Island (Innes *et al.,* 1999).

Results

Kokako productivity

The data on the numbers of male/female kokako pairs are shown in Table 1. Management intervention at most sites has clearly been associated with large increases in the numbers of female/male kokako pairs. The totals in Table 1 differ from those presented in Innes *et al.* (1999) because the latter included male/male pairs. For all of these sites the populations are not geographically closed since juveniles can disperse from the managed area (Innes *et al.,* 1999).

Note that deciding how many pairs there would have been at Mapara if there were no translocations to other sites is not as simple as saying 'since 9 females were removed 9 extra pairs might have formed'. For

Table 1. Number of female/male kokako pairs at several key sites.

Site	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
Mapara[1]	4	**6**	**6**	**10**	**15**	**17**	**18**	**31**	38	40 (49)[6]
Kaharoa		7	**10**	**12**	**18**	15	16	13	12	12
Rotoehu[2]		5	4	4	6	8	**8**	**6**		
Rotoehu[3]						10		**20**		
Otamatuna				16	12	8	13	**12**	**14**	**19**
Hunua[4]						3	2	**2**	**3**	**5**
Mataraua[5]				3	3	2	**3**	**6**	**9**	**12**

Numbers in **bold** indicate the existence of management intervention at the site, especially intensive predator control in the previous year.

[1]These tallies exclude 14 kokako which were translocated out of the block to boost populations at Tiritiri Matangi Island (1 female and 1 unknown, probably female), Pikiariki reserve (2 females and 1 male), Kapiti Island (2 females and 3 males) and the Hunua Range (4 females).

[2]These Rotoehu data were from a 150-ha study area which was selected because it had been used as a previous study area by a scientist in the 1980s. We have been informed that this was too small for accurate assessment of pest control outcomes because many juveniles dispersed outside its boundaries after fledging.

[3]In 1994 a larger survey area (450 hectares) was established and these Rotoehu data are from the larger survey.

[4]The 5 pairs include the translocations from Mapara.

[5]Pest control in Matarua was undertaken by protecting individual nests in a small block (around 440 hectares) rather than doing pest control over a larger area containing pairs.

[6]Figure in brackets for Mapara in 1998 is an estimate of the number of pairs that would have been present if the transfer from Mapara had not occurred.

example, in 1996/97 there was a translocation to Kapiti Island that included two newly-formed pairs. However, there is no accurate way of knowing whether they would have remained together and bred in that season if they had been left at Mapara. If they had remained and bred then the total pairs during 1996/97 would have been 33, and they may have produced on average one further female fledgling each. Considering that females can nest in their first year, then during the following year there might have been 38 + 2 + 2 + 5 = 47 pairs maximum, which could have led to an extra nine female fledglings. In 1998/99 there might have been a maximum of 40 + 9 + 9 = 58 pairs. Of course ecological systems do not work this simply. One kokako manager noted that as the Mapara population increases there might very well be increased competition for mates/territories and there may be a corresponding change to the time it takes for pairs to be recruited, i.e. density dependence. There may also be an increasing rate of juvenile/sub-adult mortality as population density increases. We were informed by the conservation manager that the midpoint between the maximum and minimum tallies is probably realistic if the transfers had not occurred from Mapara. For 1998, the final year of our evaluation, the mid point is 49 (see brackets in Table 1). By year, the maximum tallies were: 1996 – 33 (from 31 in Table 1 which is the minimum point), 1997 – 47 (38) and 1998 – 58 (40).

Cost effectiveness of kokako conservation

The cost effectiveness of Mapara and other kokako management sites is shown in Table 2. We report results using a 6% discount rate, and note that sensitivity tests using 3 and 10% discount rates do not significantly alter the relative cost effectiveness values for these projects. Kokako conservation has been the most cost effective at Mataraua with an average cost per additional pair of $42 976, which compares with $57 522 at Mapara and $63 094 at Otamatuna. Table 2 also shows that annual average cost per hectare is not the best measure to assess the cost effectiveness of management: Mapara costs $3 per hectare less than at Mataraua even though the cost per additional pair at Mataraua is over $14 000 less-costly than at Mapara. Conversely, management at Otamatuna was the lowest cost, at $115 per ha, while cost per additional kokako pair was highest, at $63 094.

Discussion

Subject to the caveats we discuss below, it has been possible to evaluate the cost effectiveness of kokako management at different sites with standardised data on abundance and costs. Cost effectiveness varies between sites. That knowledge should be used by

Table 2. Cost effectiveness of kokako conservation at sites. PV is the present value. K_n is the number of female/male pairs at the end of the planning period. K_i is the number of female/male pairs at the beginning of the planning period. d is the discount rate.

Site	Ha	No. years management	PV total cost d=6%	Annual amortised cost d=6%	PV of effectiveness (Kn – Ki) Pairs d=6%	Annual amortised cost per hectare d=6%	PV cost per additional kokako pair d=6%
Mataraua	440	5	$287 078	$57 416	7	$155	$42 976
Mapara	1400	12	$1 780 312	$148 359	31	$152	$57 522
Otamatuna	1300	3	$401 280	$166 962	6	$115	$63 094
Rotoehu	440		?	?	11	?	?
Kaharoa	381		?	?	4	?	?
Hunua	?		?	?	1	?	?

conservation managers to consider how best to implement the species recovery plan. Knowing that management at some sites is much more cost effective than at others should lead conservation managers to try to replicate both the natural conditions and management conditions at other places chosen to meet the plan's goals, or preferentially direct management to efficient sites. What then, are the issues that DOC and others in similar situations face in using a tool like CEA to improve their management performance?

Limitations of CEA

Cost Effectiveness Analysis has limitations and these need to be recognised by conservation managers. Where economic values are not assigned to the measure of effect, CEA will only provide results that indicate whether conservation outputs are being obtained for the least cost. As a result CEA is perhaps most useful when a decision-maker has a limited budget for a management programme such as a threatened species recovery plan and must decide what mix of actions are the most cost effective. That is the case for kokako. However, it may not be useful for informing decision-makers about the recreation, employment, income or loss of profit impacts to society of conservation management if only the direct rather than also the opportunity costs of such management are considered. For example, while CEA could be used to rank the deer control areas that provide greatest conservation return for a set budget, it says little about the potential losses (including non-monetary) endured by recreational hunters (to whom deer are a valued resource). Finally, CEA is limited by the measure of effect chosen by the analyst. If the measure of effect or output is a natural unit derived from the objective of the programme, for example, kokako male/female pairs for the kokako recovery plan, then only kokako projects can be included in the analysis. A different unit may have to be used if management is focused on multi-species. In such cases another tool, e.g. Cost Utility Analysis (see for example Cullen *et al*., 1999, 2001, 2002) or a combination of Multi Criteria Analysis and CEA (see Stephens *et al.,* 2002) need to be used.

Lack of data

The absence of cost data for Kaharoa and Rotoehu, which were two of three blocks used to test the pest limitation hypothesis (Innes *et al.,* 1999), means that it is impossible at these sites to assess the cost effectiveness of experimental management options for the kokako. Two consequences arise from the absence of expenditure data:

1. Managers and researchers have little information on the cost/uncertainty reduction trade-off (or how many financial resources it has taken to gain that extra piece of crucial knowledge). Decision-makers therefore have little ability to estimate whether more large scale and expensive adaptive management should be initiated for the kokako.
2. It is impossible to ascertain for many kokako sites how the improved knowledge base for decision-making changes the cost effectiveness ratio (i.e. cost per additional pair) in the future. Innes *et al.* (1999) do not provide data on the costs of experimental management actions at different sites.

Effectiveness data on kokako management shows that while the recovery programme has not achieved a large return in terms of the taxon (see Cullen *et al.,* 1999), it has improved the status of several populations (see Table 1). This implies the recovery plan has been successful in achieving the stated principal objective and has made progress toward the long-term goal of 1000 pairs by 2020 (Innes and Flux, 1999). Our analysis does show, however, that management has proved to be very costly. From 1987 to 1998

$2 493 839 (in 1998 dollars) was spent on kokako protection at Mapara alone, which achieved an increase of an estimated 45 male/female pairs (as at 1998). At a discount rate of 6%, the average cost per additional pair is $57 522, while the annual average cost per hectare is $116. The cost per hectare and per additional pair provide a useful benchmark for assessing how the knowledge gained (e.g. new pest control technology, identification of possum and rats as key pests) translates into improved cost effectiveness for the kokako at Mapara and Mataraua, provided data are consistently collected over time.

The cost per hectare and per additional male/female bird is high because the recovery plan aimed to simultaneously achieve research objectives (i.e. the causes of kokako decline) and management objectives (i.e. kokako improvement). However, the cost is less than that estimated for management of all New Zealand Mainland Island ecosystem restoration programmes, which was estimated by Saunders and Norton (2001) to be $165 per hectare. All else being equal, the cost of information acquisition increases relative to the magnitude of uncertainty reduction. If it were possible to separate management costs from research costs, and calculate cost effectiveness based only upon management costs, then the cost per additional pair would be somewhat less than $57 522. The more information a researcher or manager wants to gain on the dynamics of the pest resource system the greater the cost and risk involved. Before the kokako recovery programme was initiated very little was known about the cause of kokako decline and the management needs of that bird. After eight consecutive years of intensive control of possums, ship rats, goats (*Capra hircus*), stoats (*Mustela erminea*), weasels (*Mustela nivalis*), ferrets (*Mustela furo*) and cats (*Felis catus*), it has been deduced that not all of these pests need to be controlled every single year for kokako protection (Innes *et al.,* 1999). Conservation managers wishing to recover kokako populations should aim for residual densities of 1 possum per 100 trap nights and a 1% ship rat tracking index at the onset of the nest season each year for several years (Innes *et al.,* 1999). From this perspective there is reason to expect the cost per additional pair may be less than $57 522 in the future. Due to the required intensity and frequency of rat and possum control mainland kokako management will continue to be costly, however. Just how costly kokako recovery will be in the future is dependent on the pulsing intervals of pest control, future improvements in pest control technology, and more importantly how realistic the stated recovery goal is for the 21st century.

Project goal selection

The 20-year goal in the updated draft kokako recovery plan is to restore the national population to 1000 pairs by 2020 in sustainable communities throughout the North Island, or more precisely 50 pairs at each of 20 sites (Innes and Flux, 1999). No definition of what is meant by 'sustainable communities' is provided. Experimental management for the kokako has shown the immediate cause of decline is recruitment failure due to predation by ship rats and possums at kokako nests and that the decline can be reversed, by intensive and sustained pest control using existing technology (Innes *et al.,* 1999; Innes and Flux, 1999). However, the proposed recovery goal seems to have been determined without reference to the economic dimension of the relationship between pests, people and valued resources (Parkes, 1993). An *ad hoc* approach to goal determination implicitly assumes the conservation budget is always large enough to fund any vision that can be conceived of and that the opportunity costs of kokako conservation are zero. This stance is clearly at odds with some of DOC's more recent statements including … 'rigorous prioritisation in relation to available funding occurs … Prioritising will be guided by (vi) … cost effectiveness' (Department of Conservation, 2001).

Recent work by Stephens *et al.* (2002) for the Department does recognise the importance of costs. Measuring Conservation Achievement (MCA) is a methodology to inform selection of asset protection and recovery actions. The goal is to develop a model of the expected contribution of each conservation project to maintenance or enhancement of mainly native vegetation and other native species. The ultimate goal of conservation is argued to be maintenance or improvement of 'condition' — compositional similarity to the biota which it is believed would occur at a site if it did not suffer from human-induced disturbance. The Stephens *et al.* (2002) approach is to measure the effects of management on for example pests, and to estimate possible conservation outcomes via explicit relationships between proximate targets and 'condition'. As far as its current application is concerned, however, MCA deals with site selection prioritisation and not with project choices within a single-species recovery programme. Notwithstanding this criticism, MCA appears to have the longer-term potential to contribute to such deliberations.

In evaluating the desirability of the management goal for a species such as the kokako it is necessary to examine the marginal trade-offs between a range of population sizes or survival probabilities and costs (Hyde, 1989; Haight, 1995; Montgomery *et al.,* 1994). In general, the more managers and researchers aim to improve the survival probability of a species or population, the greater the cost. Montgomery *et al.* (1994) show how the cost of more ambitious conservation plans for the northern spotted owl (*Strix occidentalis caurina*) increases with successive increments of survival probability. The authors show that the marginal cost of improving the survival probability by one percentage point from 90 to 91% is US$1.4 billion. To improve the owl's survival probability a further 5% to 95% will cost on average an additional $2.6 billion per percentage point (Montgomery *et al.,* 1994). Haight (1995) explores the tradeoff between target population size and foregone timber revenue, and finds results which are qualitatively similar, although in much smaller magnitudes, to those of Montgomery *et al.* (1994).

The merit of the goal of the North Island kokako recovery plan compared with many other options is not known because the costs and trade-offs have not been evaluated in a similar manner. The costs of moving from approximately 270 to 1000 kokako pairs are not known. There is no information available to indicate the relationship between increases in population size and the marginal cost of improving the survival probability of the kokako. If the goal of 1000 pairs is very costly, a more productive use of scarce resources may be to aim for fewer pairs and use the remaining resources for the brown kiwi, black stilt, or other threatened species. Such questions cannot be answered by ecological science, which as Baskerville (1997) points out, 'too frequently assumes there are no costs to be borne from ecological action'. Since economic parameters appear to be mostly absent in the recovery plan, if it is implemented poor resource allocation decisions may eventuate.

Legislative and policy issues with economic evaluation

It is impossible to accurately assess the cost effectiveness of programmes if there are no accurate expenditure data. It is also difficult to estimate how much has been spent on species recovery programmes if several objectives are pursued at a site. There are other factors that help explain the difficulty in cost estimation. The Conservation Act 1987, National Parks Act 1980, Wild Animal Control Act 1977, Reserves Act 1977 and the Wildlife Act 1953 do not require an evaluation of costs and benefits in decision-making. This is unlike other environmental laws such as the Resource Management Act 1991 (s.32) and the Biosecurity Act 1993 (s.57), which require an analysis of costs and benefits. There is some mention of costs and benefits in the Public Finance Act 1989, but only in a very limited way in Part I section 9. The absence of a requirement for economic evaluation of conservation management means costs can be ignored, or at worst, assumed to be zero. However, as already noted, DOC have taken some steps in addressing this issue (Stephens *et al.,* 2002) and are now in the process of slowly implementing their system.

The Department of Conservation's existing priority setting systems for threatened species, introduced mammals and weeds do not explicitly incorporate the direct costs of achieving conservation goals. They mostly refer to the biological parameters of the resource allocation problem, with some minor attention to human values as in Molloy and Davis (1994). Although there is some mention of other factors including financial aspects, which are meant to be assessed in a 'separate exercise' (Molloy and Davis 1994), there is no reference given to the details of this exercise and how this information will be used in conjunction with the ranking system. While it is appropriate that initial prioritisation occurs on the basis of ecological criteria, the current system implies that the direct costs of conservation or pest control are not relevant in ranking management actions for threatened species or pest control sites. An outcome is that conservation managers have no real need for accurate cost data in allocating conservation resources, thus perpetuating the information shortfall and inability to properly consider trade-offs.

Current guidelines for writing species recovery plans do not require budget estimates. The only mention of budgeting responsibilities is for things such as the 'cost of venue and food if a recovery group meeting is necessary', and 'the cost of publishing a plan' (Department of Conservation, 1999b). Cost data that could be quantified include the alternative courses of recovery action, e.g. captive rearing, pest and weed control, fencing and advocacy. The estimation of programme costs is left up to conservancies who may only be pursuing one particular aspect of a programme, which may cut across several conservancies (Craig, 1997). This approach implicitly assumes the conservation budget is large enough for recovery groups to achieve any goal they want and that conservancies will always have enough resources to carry out the vision of the recovery group.

Accounting systems are often designed primarily for record keeping and not for policy analysis purposes. There is no clear link among stated recovery goals, DOC's output classes, and the data recorded in the DOC accounting system. Money is available for threatened species management in different output classes, which essentially have similar outcomes. Output class 4.2 covers animal and plant pest control and Output class 5.1/5.2 covers species conservation programmes.

Concluding comments

Craig (1997) has severely criticised DOC's approach to costing threatened species recovery programmes. He argued that the present cost-oblivious approach to conservation is one of the reasons why the 'New Zealand government heavily constrains finances to the

Department of Conservation' (Craig, 1997). More recently, and certainly in the last few budgets, there has been a substantial increase in government investment in threatened species conservation, although this is often labelled as biodiversity or ecosystem conservation. However, the need for increased use of economic analysis to guide conservation decision-making remains an important issue.

We argue here and elsewhere that economics can contribute to conservation decision-making (Hughey *et al.,* 2003). Two key questions conservation managers need to ask are (1) where should scarce conservation resources be invested?, and (2) which investments in conservation management have been best? Cost effectiveness analysis and cost utility analysis are necessary tools to answer those questions, but accurate cost data are essential for both techniques to be usable. These two key questions are more likely to be posed and answered if conservation decision-makers are required to focus directly on the economic implications of conservation decisions.

Acknowledgements

The authors thank the following for their assistance and advice: Jim Coleman, Graham Nugent, John Innes (all of Landcare Research New Zealand Ltd.), Paul Jansen and individual kokako project site managers from the Department of Conservation, and two referees for this journal. Funding for this research was provided by the Foundation for Research, Science and Technology, via sub-contract LIN 1156 with Landcare Research New Zealand Ltd.

References

Baskerville, G.L. 1997. Advocacy, science, policy and life in the real world. *Conservation Ecology [online] 1:* 9 Available from the Internet. URL: http://www.consecol.org/vol1/iss1/art9/index.html

Craig, J.L. 1997. Managing bird populations: for whom and at what cost? *Pacific Conservation Biology 3:* 172-182.

Cullen, R.; Fairburn, G.A.; Hughey, K.F.D. 1999. COPY. A new technique for evaluation of biodiversity protection projects. *Pacific Conservation Biology 5:* 115-123.

Cullen, R.; Fairburn, G.A.; Hughey, K.F.D. 2001. Measuring the productivity of threatened species programs. *Ecological Economics 39:* 53-66.

Cullen, R.; Moran, E.; Hughey, K.F.D. 2002. Evaluation of six multiple species projects in the conservation of threatened species. *Conservation Science Newsletter 45:* 11-13.

Department of Conservation. 1999a. *Department of Conservation annual report for the year ended 1999*. Department of Conservation, Wellington, N.Z.

Department of Conservation. 1999b. *Standard operating procedure – species recovery planning*. QD code: NH1043/03, Department of Conservation, (Mimeo) Wellington, N.Z.

Department of Conservation. 1998. *Restoring the dawn chorus: Department of Conservation strategic business plan 1998-2002*. Department of Conservation, Wellington, N.Z.

Department of Conservation. 2001. *Restoring the dawn chorus 2001-04*. Department of Conservation, Wellington, N.Z.

Goldstein, P.Z. 1999. Functional ecosystems and biodiversity buzzwords. *Conservation Biology 13:* 247-255.

Haight, R.G. 1995. Comparing extinction risk and economic cost in wildlife conservation planning. *Ecological Applications 5:* 767-775.

Hitchmough, R. 2002. *New Zealand threat classification system lists 2002*. Threatened Species Occasional Publication 23. Department of Conservation, Wellington, N.Z.

Hughey, K.F.D.; Cullen, R; Moran, E. 2003. Integrating economic approaches into the evaluation of conservation management initiatives. *Conservation Biology 17:* 93-103.

Hyde, W.F. 1989. Marginal costs of managing endangered species: the case of the red cockaded woodpecker. *Journal of Agricultural Economics Research 41:* 12-19.

Innes, J.; Flux, I. 1999. *North Island Kokako recovery plan 1999-2009*. Third edition, July 1999, Threatened Species Recovery Plan 30. Department of Conservation, Wellington, N.Z.

Innes, J.; Hay, J.R.; Flux, I.; Bradfield, P.; Speed, H.; Jansen, P. 1999. Successful recovery of North Island kokako *Callaeas cinerea wilsoni* populations, by adaptive management. *Biological Conservation 87:* 201-214.

Metrick, A.; Weitzman, M.L. 1998. Conflicts and choices in biodiversity preservation. *Journal of Economic Perspectives 12(3):* 21-34.

Ministry for the Environment. 1994. *Environment 2010 strategy*. Ministry for the Environment, Wellington, N.Z.

Molloy, J.; Davis, A. 1994. *Setting priorities for the conservation of New Zealand's threatened plants and animals*. Department of Conservation, Wellington, N.Z.

Montgomery, C.A.; Brown, G.A.; Adams, D.M. 1994. The marginal cost of species preservation: the northern spotted owl. *Journal of Environmental*

Economics and Management 26: 111-128.

Moran, D.; Pearce, D. 1998. The economics of biodiversity. *In:* Folmer, H.; Tietenberg, T. (Editors), *The international yearbook of environmental and resource economics 1997/98 — A survey of current issues*, pp. 82-113, Edward Elgar Publishing, U.K.

Noss, R.F. 1990: Indicators for monitoring biodiversity: a hierarchical approach. *Conservation Biology 4:* 355-364.

Parkes, J.P. 1993. The ecological dynamics of pest-resource-people systems. *New Zealand Journal of Zoology 20:* 223-230.

Editorial Board member: Doug Armstrong

Parkes, J.P. 1996. Integrating the management of introduced mammal pests of conservation values in New Zealand. *Wildlife Biology 2:* 179-184.

Saunders, A. 1999. *Mainland islands – a review*. Draft for discussion, Department of Conservation, (Mimeo) Wellington, N.Z.

Saunders, A.; Norton, D.A. 2001. Ecological restoration at mainland islands in New Zealand. *Biological Conservation 99:* 109-119.

Stephens, T.; Brown, D.; Thornley, N. 2002. *Measuring conservation achievement: concepts and their application over the Twizel Area*. Science for Conservation Series 200. Department of Conservation, Wellington, N.Z.